

**A Model for the Assessment of Negligible Impact
for Incidental Take of Manatees
due to Watercraft-related Activities**

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A stochastic matrix-based model of Florida manatee (*Trichechus manatus latirostris*) population dynamics will be used to make a preliminary assessment of levels of watercraft-related incidental take that can be considered negligible under the Marine Mammal Protection Act (MMPA). To be considered negligible, “The impact cannot be reasonably expected to, and is not reasonably likely to, adversely affect the species through effects on annual rates of recruitment or survival.” (MMPA 50 C.F.R. 18.27). Several principles underlie the development of this model: (a) it will be based on the best currently available information about manatee population dynamics; (b) it will explicitly incorporate uncertainty about dynamics and population parameters, and where the uncertainty cannot be made mathematically explicit, precautionary assumptions will be made; and (c) it will meet the standards specified by, implied by, or interpreted from the MMPA.

The purpose of the model is to calculate, through stochastic simulations, whether a given set of authorized take levels in each of the four regions of the Florida manatee is expected to meet the criteria for having a negligible impact. Thus, the input for the model is a set of 4 values (one value for each region) for annual authorized take, as measured by observed watercraft-related deaths. The outputs of the model are: the probabilities of having achieved recovery (defined below) within 50 and 100 years, given the proposed level of incidental take; and the probability of recovery being delayed by $\geq 10\%$ with the proposed level of take, compared to the case where there was no take. For a proposed level of take to be considered negligible, these three probabilities should be $\geq 95\%$, $\geq 99\%$, and $\geq 95\%$, respectively. These criteria are based on legal precedent set with regard to the MMPA.

To determine whether the simulated population has achieved recovery, the recovery criteria in the 3rd revision of the Florida Manatee Recovery Plan (USFWS 2001) will be used, namely, simultaneous achievement in all four regions of (a) statistical confidence (of 95%) that the twenty-year average annual survival rate for adult manatees is ≥ 0.90 ; (b) statistical confidence (of 95%) that the twenty-year average annual fraction of adult female manatees accompanied by first or second year calves in winter is ≥ 0.40 ; and (c) statistical confidence (of 95%) that the corresponding model-based annual growth rate is nonnegative.

The model will be used in two ways: first, to determine whether current levels of incidental take meet the criteria for negligible impact; and second, through iterated simulations, to find the maximum levels of authorized take in each region that still meet the criteria for negligible impact.

This model will be built from the best *available* knowledge, data, models, and analyses. A separate 3-5 year effort is beginning to *develop* a more comprehensive model that integrates the multiple annual sources of information available about manatees.

Model Structure

The core of the model (Fig. 1) is a matrix-based description of *female* manatee population dynamics. The model centers on females because their survival and reproduction directly control population growth. Manatees have a promiscuous mating system. A single male can inseminate multiple females (Hartman 1979:100); therefore males do not directly limit population growth. The core model is expanded below to include males.

In the core model, the manatee population is broken into 7 classes of females:

First-year calves (0.5-yr-old). Manatee population monitoring focuses on the winter aggregation sites. Calves, however, are born during the spring and summer (Marmontel 1995:105, Rathbun et al. 1995, O'Shea and Hartley 1995, Reid et al. 1995). The first reliable data on reproduction is collected when a female with a dependent nursing calf returns to the winter aggregations in fall and winter. Calves are ca. 3-9 months old at this time or a mean of 0.5 yr. Thus a first-year calf

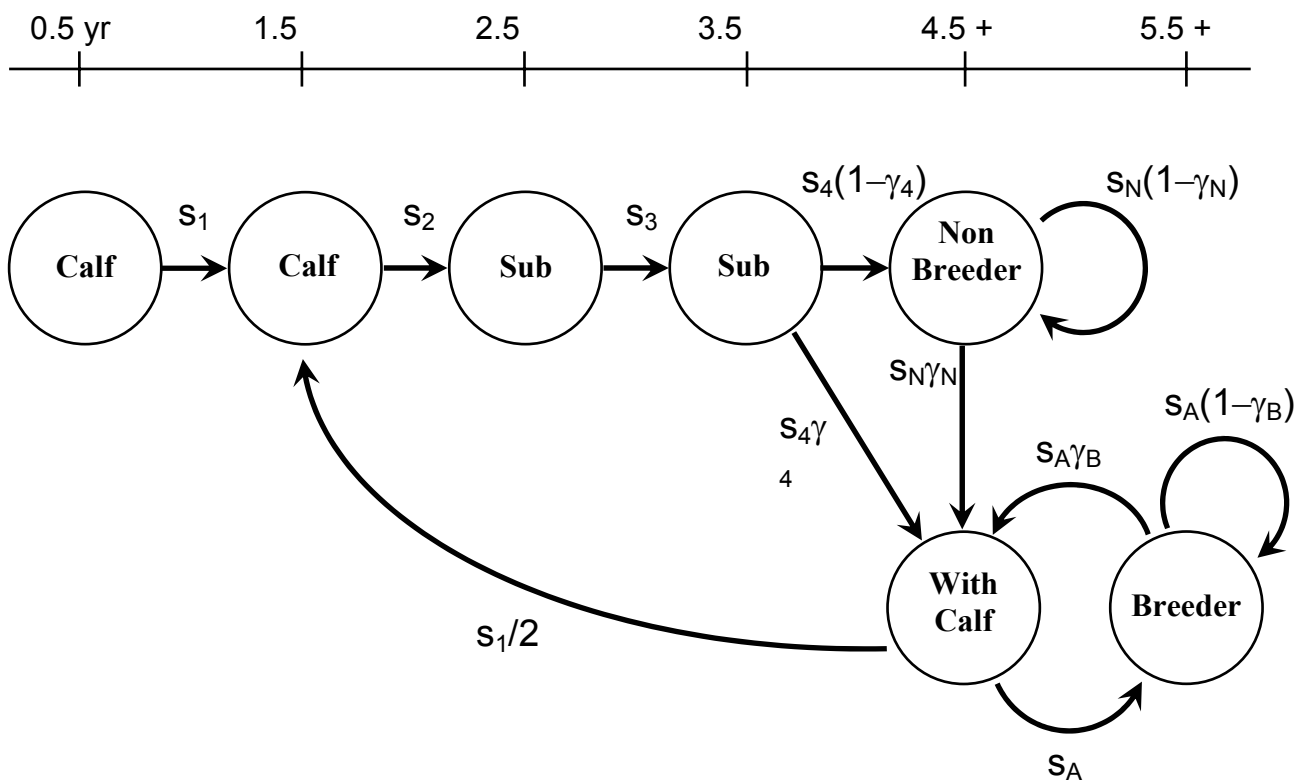


Fig. 1. Life-history diagram for the manatee population model. Note that calves enter the population as separate entities at 1.5 yr. The first circle is shown for completeness.

represents successful pregnancy, birth, and survival to ca. age 0.5. There currently are no reliable means to monitor pregnancy or births (Rathbun et al. 1995:142).

Second-year calves (1.5-yr-old). Data on second-year calves are collected the following year at the aggregation site. Second year calves are primarily identified by size—they are larger than first-year calves, but smaller than subadults. They may or may not be weaned and independent of their mothers. There is considerable variation among individuals as to whether a calf will nurse for one or two years (Rathbun et al. 1995:145, O’Shea and Hartley 1995:161, Reid et al. 1995:183).

Third-year subadults (2.5-yr-old). At three years of age, individuals are independent but only rarely sexually mature and capable of reproducing (Marmontel 1995:103, O’Shea and Hartley 1995:162).

Fourth-year subadults (3.5-yr-old) and Non-breeders (≥ 4.5 -yr-old). Non-breeders are individuals 4.5 years old or older that have not yet successfully reproduced. This model assumes that the earliest a female can breed is in her fourth year (at age ~ 3.5 yr), thus, the earliest first parturition can occur is age 4.5 yr. Based on winter observations, the earliest that a female manatee has been observed with a dependent calf is four winters after she herself was observed as a new calf, that is, at ca. 4.5 yr (Rathbun et al. 1995, O’Shea and Hartley 1995). However there is considerable individual variation in the age of first successful reproduction (Marmontel 1995:103; O’Shea and Hartley 1995:162); this is reflected in females that remain in the non-breeder class for some time.

Adults with first-year calves and Breeders. Sexually mature females that are accompanied by a dependent first-year calf, or that have previously produced a calf are classified as “with a 1st-yr calf” or as a “breeder,” respectively. Mature females accompanied by a not-yet-weaned second-yr calf are considered breeders, since the attendant calf was not born during the current year. Note that there is no class for “old” adults, since there is no evidence for senescence in manatees (Marmontel 1995).

Two types of life-history parameters describe the transitions between the classes in the model: survival rates (s) and breeding rates (γ). For instance, s_1 is the probability a first-year calf survives to become a second-year calf; γ_N is the probability that an adult female that has not yet given birth to a calf, breeds and successfully gives birth within the next year. Non-breeders that survive either give birth to a calf (with probability γ_N) or remain as non-breeders. Females with a first-year calf that survive become breeders the next year (with probability = 1.0), regardless of whether they wean the calf after the first year. That is, the model does not allow for females to have calves two years in a row—this constraint reflects the physiological limitations imposed by the length of pregnancy (12-13 months, Rathbun et al. 1995, O’Shea and Hartley 1995, Reid et al. 1995) and early dependence of the calf. Breeders that survive to the next year either give birth to a calf (with probability γ_B) or remain as breeders. A female with a first-year calf gives rise to a second-year calf (weaned or not weaned) in the next year with probability $s_1/2$, reflecting the probability of calf survival and an even primary sex ratio (recall this is a model for the female segment of the population, and only half the calves are expected to be female). Note

that in this model, the litter size is assumed to be 1 calf. While twinning is possible in nature, it is rare (Marmontel 1995, Rathbun et al. 1995, O'Shea and Hartley 1995).

This life history diagram (Fig. 1) can be expressed in matrix form as

$$\begin{bmatrix} N_2 \\ N_3 \\ N_4 \\ N_{NB} \\ N_C \\ N_B \end{bmatrix}_{t+1} = \begin{bmatrix} 0 & 0 & 0 & 0 & \frac{1}{2}s_1 & 0 \\ s_2 & 0 & 0 & 0 & 0 & 0 \\ 0 & s_3 & 0 & 0 & 0 & 0 \\ 0 & 0 & s_4(1-\gamma_4) & s_N(1-\gamma_N) & 0 & 0 \\ 0 & 0 & s_4\gamma_4 & s_N\gamma_N & 0 & s_A\gamma_B \\ 0 & 0 & 0 & 0 & s_A & s_A(1-\gamma_B) \end{bmatrix} \begin{bmatrix} N_2 \\ N_3 \\ N_4 \\ N_{NB} \\ N_C \\ N_B \end{bmatrix}_t \quad (1)$$

where the N_i represent the number of manatees in class i at a given point in time. In the matrix formulation, first-year calves are not counted separately, as they are assumed to be dependent on their mothers, although their numbers can be inferred from the number of females with calves (N_C). New births first appear in the population model as second-year calves. The total female population size at time t can be calculated as:

$$N_{Total} = N_2 + N_3 + N_4 + N_{NB} + 1.5N_C + N_B \quad (2)$$

where the number of females with first-year calves is multiplied by 1.5 to include both the mothers and their *female* calves in the total.

To expand the core model to include males, four additional classes of animals are added. The matrix formulation is

$$\begin{bmatrix} N_2 \\ N_3 \\ N_4 \\ N_{NB} \\ N_C \\ N_B \\ N_2^M \\ N_3^M \\ N_4^M \\ N_A^M \end{bmatrix}_{t+1} = \begin{bmatrix} 0 & 0 & 0 & 0 & \frac{1}{2}s_1 & 0 & 0 & 0 & 0 & 0 \\ s_2 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & s_3 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & s_4(1-\gamma_4) & s_N(1-\gamma_N) & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & s_4\gamma_4 & s_N\gamma_N & 0 & s_A\gamma_B & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & s_A & s_A(1-\gamma_B) & 0 & 0 & 0 & 0 \\ \hline 0 & 0 & 0 & 0 & \frac{1}{2}s_1 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & s_2 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & s_3 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & s_4 & s_A \end{bmatrix} \begin{bmatrix} N_2 \\ N_3 \\ N_4 \\ N_{NB} \\ N_C \\ N_B \\ N_2^M \\ N_3^M \\ N_4^M \\ N_A^M \end{bmatrix}_t \quad (3)$$

where the four new classes keep track of the second-year, third-year, fourth-year, and adult males, respectively. There is no evidence that survival rates differ between males and females

(Langtimm et al. 1998), so the same survival rates are used for corresponding male and female age-classes. The total population size can be written as

$$N_{Total} = N_2 + N_3 + N_4 + N_{NB} + 2N_C + N_B + N_2^M + N_3^M + N_4^M + N_A^M \quad (4)$$

with the females with first-year calves now multiplied by two to include both the mothers and their calves (of either sex) in the total.

Environmental Stochasticity

Variation in life-history parameters (survival and reproductive rates) due to uncontrolled factors in the environment is called environmental stochasticity. Manatees experience environmental stochasticity due to a number of factors, for example, red tide (O'Shea et al. 1991, Bossart et al. 1998), severe cold (Buergelt et al. 1984), and hurricanes (Langtimm and Beck, *in press*). In this model, specific sources of variation are not treated separately, but instead their combined effect is included in an aggregate measure of variation. That is, catastrophes are not distinguished from "normal" variation. The time series of observations used to estimate survival and reproduction include years when these factors were operating, and so the estimated life-history parameters integrate stochasticity from all sources.

In this model, environmental stochasticity is represented by probability distributions for the annual values for the life-history parameters. All of the parameters in the model are probabilities (survival probabilities, s ; breeding probabilities, γ) and thus must be in the interval $[0,1]$. Variation in these parameters is modeled with beta distributions. Like the normal distribution, the beta distribution is a two-parameter probability distribution; but unlike the normal distribution, the random values of the beta distribution are confined to $[0,1]$, so it is natural to use this distribution for parameters like survival rate.

Specification of the beta distribution for each life-history parameter requires a mean value, μ , and a concentration parameter, θ . The concentration parameter reflects how "tight" the distribution is—a larger value of θ results in narrower probability distribution, hence a smaller variance (the variance of the beta distribution is given by $\mu(1-\mu)/(\theta+1)$). The year-specific values for each life-history parameter are sampled from the appropriate beta distribution. Thus, the first-year calf survival rate in year t has the distribution

$$s_{1,t} \sim \text{beta}(\mu_{s_1}, \theta_{s_1}). \quad (5)$$

The survival rates are assumed to vary together, since it is likely the same environmental factors will affect subadult survival rates as affect adult survival rates. To model this, the adult survival rate is sampled first. Then the cumulative probability value is found for the observed adult survival rate, and this cumulative probability value is used to find the corresponding value for the other survival rates. This method produces perfect temporal correlation among the survival rates sampled in this manner.

For example, suppose for s_A , the mean is 0.94 and the concentration parameter is 50 (this is equivalent to a standard deviation of 0.033); and for s_I , the mean is 0.82 and the concentration parameter is 20 (sd = 0.084). The adult survival rate is drawn from the appropriate beta distribution. Suppose this value is 0.91, the cumulative probability for which is 0.17 (i.e., 17% of the values drawn for adult survival will be less than 0.91). The corresponding value for first-year calf survival below which 17% of the values will fall is 0.74. Thus, the adult and first-year calf survival rates would be 0.91 and 0.74, respectively, for that particular year. This same method can be applied to a larger set of survival rates.

The breeding probabilities, γ , are also assumed to vary together, but are assumed to be independent of the survival rates. This implies that the same set of environmental factors affects all three breeding probabilities (γ_{NB} , γ_C , and γ_B), but this set of environmental factors is different from the factors that affect annual survival.

Demographic Stochasticity

Demographic stochasticity is the variation due to applying probabilistic life-history parameters to individuals. For example, suppose the survival rate in a population is 0.5. If there are 10 animals in the population, 5 are expected to survive, but the number that actually survive can vary, since each animal flips its own “survival coin.” Since all the life-history parameters in the model are probabilities with binary outcomes (survive vs. not-survive, breed vs. not-breed), an appropriate distribution for the demographic stochasticity is the binomial distribution (the binomial is the “coin-flip” distribution—e.g., if I flip 100 weighted coins, each with a certain probability of landing heads, what’s the probability that, say, 65 of them are heads?). Thus, for each class in the population model, the number that survive or breed is drawn from a binomial distribution with success probability equal to the year-specific value for the appropriate life-history parameter.

For example, suppose in a given year, there are 100 adult males; the mean adult male survival rate is 0.94, and the concentration parameter is 50. The year specific annual survival rate is

$$s_{A,t}^M \sim \text{beta}(0.94, 50).$$

Let’s say a draw is made from this distribution and the value is 0.88. Then, the actual number of adult males that survive has the distribution

$$\text{binomial}(100, 0.88).$$

Let’s say a draw is made from this distribution and the value is 91. Thus, of the 100 adult males alive in year t , 91 survived to the next year.

Because demographic stochasticity represents the application of life-history parameters to individuals, it is calculated independently for each class in the model (this is equivalent to calculating it independently for each individual in the population).

The sex ratio in the first-year calves is assumed to be 0.50, there being no evidence of a skewed primary sex ratio or differential neonatal survival by sex (O'Shea and Hartley 1995, Reid et al. 1995). The number of first-year calves is determined by the number of females with calves. The number of *female* first-year calves is sampled from a binomial distribution with success probability 0.50. The number of male first-year calves is then found by subtraction.

Parameter Estimates

The parameter estimates used in the model are shown in Table 1. The values in bold are direct estimates found either in the published literature or in more recent analyses. The values in plain text are inferred, and the rationale is explained in the footnotes. Missing values are parameters for which direct estimates do not yet exist and which will require additional analyses or carefully reasoned assumptions.

As a part of the review of this document, several expert panels will be convened in early 2003 to provide critique of the model structure and parameter estimates, and guidance on how to estimate the missing values.

Initial Population Size and Structure

The synoptic survey count from January 5-6, 2001 is used as the baseline for all the simulations (Table 2, for description of the surveys see Ackerman 1995). While there is substantial disagreement about whether these counts represent an unbiased estimate of the current population size, there is agreement that these counts at least represent minimum population sizes (Ackerman 1995). In keeping with the precautionary approach of the MMPA, then, this model uses the count for each Region as a conservative estimate of the initial population size.

For each region, the expected population structure (the fraction of the population in each sex/stage class) is found from the stable stage distribution. In matrix population models, the fraction of animals in each stage class is known to stabilize after some period of time, even if the population continues to grow or decline (Caswell 2001). This set of fractions is called the stable stage distribution, and is found mathematically from the dominant eigenvector of the projection matrix. In this model, the expected initial population structure is the eigenvector of the projection matrix found in equation (3), using the mean values for each life-history parameter. This expected population structure is then used as the set of probabilities in a multinomial distribution. For each replicate of a simulation, the initial population structure is drawn randomly from this multinomial distribution, with the total population size given in Table 2. This means that the starting population size will be the same for all replicates in the simulation, but the population structure will vary, to reflect uncertainty about the actual structure of the population in 2001.

Density-dependence

The model shown in equation (3) is an exponential population model, and the addition of stochastic effects doesn't change that. No real population can grow exponentially for an

Table 1. Parameter estimates for Florida manatees in the four sub-populations. Values in **bold** are direct estimates of the appropriate parameter from published studies or recent analyses. Values in roman type are inferred. Values not given will need to be determined from new analyses or well-reasoned assumptions. The “uncertainty” column represents a range of potential values for each parameter; in general, this is the 95% confidence interval for the parameter estimate.

Parameter	Atlantic		Upper St. John's		Northwest		Southwest	
	Estimate	Uncertainty	Estimate	Uncertainty	Estimate	Uncertainty	Estimate	Uncertainty
<i>Means</i> (μ)								
s_1			0.810^a	(.727, .873)	0.810 ^l	(.7, .9)		
s_2			0.915^a	(.827, .960)	0.915 ^l	(.8, .96)		
s_3	0.943 ^k	(.923, .962)	0.961^a	(.915, .983)	0.961 ^l	(.9, .98)	0.906 ^k	(.867, .944)
s_4	0.943 ^k	(.923, .962)	0.961^a	(.915, .983)	0.961 ^l	(.9, .98)	0.906 ^k	(.867, .944)
s_N	0.943^b	(.923, .962)	0.961^c	(.900, .985)	0.962^d	(.953, .972)	0.906^e	(.867, .944)
s_A	0.943^b	(.923, .962)	0.961^c	(.900, .985)	0.962^d	(.953, .972)	0.906^e	(.867, .944)
γ_4			0.208^f	(.071, .422)	0.000^g	(.000, .285)	0.0 ^m	(.0, .3)
γ_{NB}			0.610^f	(.505, .709)	0.381^g	(.181, .616)	0.304^h	(.132, .529)
γ_B			0.610^f	(.505, .709)	0.526ⁱ	(.414, .638)	0.595^h	(.421, .752)
<i>Concentration</i> (θ)								
s_1								
s_2								
s_3	51 ^k	(13, 113)	350 ^k	(25, 1000)	344 ^k	(25, 1000)	100 ^k	(5, 300)
s_4	51 ^k	(13, 113)	350 ^k	(25, 1000)	344 ^k	(25, 1000)	100 ^k	(5, 300)
s_N	51^j	(13, 113)	350 ⁿ	(25, 1000)	344^j	(25, 1000)	100^j	(5, 300)
s_A	51^j	(13, 113)	350 ⁿ	(25, 1000)	344^j	(25, 1000)	100^j	(5, 300)
γ_4								
γ_{NB}								
γ_B								

^a Langtimm, C.A. Unpublished analysis, 3/26/02, of known-age individuals at Blue Spring State Park. The data and estimates are essentially the same as in O'Shea and Hartley (1995), but formal mark-recapture methods were used to obtain the estimates and confidence intervals.

^b Langtimm, C.A. Unpublished analysis, 3/26/02, of Atlantic coast photo-ID data, 1984-2000. Here, and elsewhere, we assume that the survival rates of adults does not depend on sex or breeding status.

^c Langtimm et al. (1998).

^d Langtimm, C.A. Unpublished analysis, 3/26/02, of Crystal River photo-ID data, 1981-2000.

^e Langtimm, C.A. Unpublished analysis, 9/9/02, of Southwest photo-ID data, 1994-2001.

^f Runge, M.C. Unpublished analysis, 3/27/02, of reproductive histories of known-age females at Blue Spring.

^g Runge, M.C. Unpublished analysis, 3/27/02, of reproductive histories of known-age females at Crystal River.

^h Runge, M.C. Unpublished reanalysis, 9/11/02, of reproductive histories from Sarasota Bay (Koelsch 2001).

ⁱ Kendall et al. (2002, in review), modified to include a longer set of data, 1982-2000.

^j Runge, M.C. and C.A. Langtimm. Unpublished analysis, 9/10/02, to estimate parameters of a beta distribution from the mean and temporal variance of survival based on photo-ID data.

^k Parameter estimates for s_3 and s_4 , were assumed to be the same as corresponding values for adults. Subadult survival rates have not been shown to differ from adult survival rates (O'Shea and Hartley 1995).

^l Calf and subadult survival rates for the Northwest region were assumed to be similar to the corresponding rates in the Upper St. John's region, since these two populations are both well protected and healthy. The confidence intervals were increased slightly to reflect the additional uncertainty associated with this assumption.

^m Koelsch (2001) did not study known-age individuals, so estimates of γ_4 were not possible. But the resemblance between the reproductive rates in Sarasota Bay and Crystal River suggests this rate might also be similar.

ⁿ The USJ and NW regions were assumed to be similar with regard to the temporal variance of survival.

Table 2. Regional counts from the synoptic aerial survey, January 5-6, 2001.

Region	Count
Atlantic	1408
Upper St. John's River	112
Northwest	377
Southwest	1379
<i>Total</i>	<i>3276</i>

indefinite period of time—at some point, some resource becomes limiting and survival and/or reproductive rates must decrease.

Density effects on life-history parameters in manatees have not yet been documented or estimated in the literature. Four possible reasons for this are: (1) manatee densities may be too low to have shown any strong density-dependent effects; (2) since robust monitoring programs have been in place, manatee densities may not have varied over a wide enough range to allow detection of density-dependent effects; (3) appropriate monitoring programs specifically geared to detect density-dependent effects have not been developed; and (4) the relevant limiting factors may be unknown and/or may change over time and space, making detection of the effects of limitation difficult.

One of the major limiting factors for manatee population growth is presumed to be warm-water refugia (USFWS 2001). As the older power plants that currently provide warm-water are phased out of use, it is possible that manatee populations, particularly in the Atlantic and Southwest regions, will experience a reduction in the carrying capacity of their environments. In addition, reduction of spring flows due to increasing human reduction of aquifer capacity is decreasing the availability of warm-water at natural springs. Whether these factors affect long-term recovery of manatees will depend on the extent to which this loss of warm-water can be mitigated by other management measures.

To reflect uncertainty about the effects of density-dependence, particularly as related to the winter warm-water carrying capacity, three possible scenarios will be considered. (1) No density-dependence over the time frame of the simulations. This could be the case if manatees can adapt to changes quickly enough, or if management actions can be swift enough, to effectively increase the carrying capacity faster than any increase in the population size. (2) Fixed carrying capacities. Current winter warm-water carrying capacities are assumed to remain constant over the time frame of the simulations. This would be the case if there were no changes to warm-water sites, or if moderate mitigating measures were implemented. (3) Declining carrying capacity over time. This scenario would reflect substantial loss of warm-water refugia over the next 50 years, due to closure of power plants and reduced spring flow.

The latter two scenarios are implemented in the model by including an additional source of mortality in cold years for that portion of the population that exceeds the winter carrying capacity. The details of how to determine the current carrying capacity, the rate of decline of that carrying capacity, and the mortality rates associated with being outside warm-water refugia during cold spells will be specified by a convened expert panel in early 2003.

To produce a single result from the simulations, rather than consider these three scenarios separately, each is assigned a probability of occurrence. For each replicate of a simulation, the scenario is selected according to these probabilities of occurrence. The same expert panel will participate in setting these probabilities. As with other parameters in the model, the uncertainty associated with these scenarios will be reflected in a range of probabilities for each.

The Effect of Watercraft-related Mortality

The crux of the model is the link between the life-history parameters and the number of watercraft-related mortalities observed in the carcass recovery program each year. (See Ackerman et al. 1995 for a description of this program.)

The survival estimates shown in Table 1 include the effects of watercraft-related mortality in those regions over the time frame the estimates were made. To estimate the region-specific survival rates in the absence of watercraft-mortality, the mortality rates will be decreased proportionally by the fraction of carcasses for which the cause of death is determined to be watercraft-related. That is, if 35% of the carcasses were due to watercraft-related-mortality, then the observed survival rate would be increased 35% of the way toward 1.00. Regional and age-specific carcass data for the last 10 years will be used to estimate the fractions of mortality due to various causes.

The “undetermined” cause of death category raises a slight problem in that some of these deaths may be due to watercraft. However, necropsy experts estimate that perhaps only 1-2 % of the “undetermined” causes of death might be due to watercraft, because the extreme trauma associated with a watercraft death (e.g., broken bones, massive cutting) is typically detectable even in badly decomposed carcasses (Robert K. Bonde, *personal communication*). To reflect uncertainty in the proportion of “undetermined” carcasses that are actually watercraft related deaths, a 95% confidence interval of 0-10% for this proportion was used in calculating the survival-in-the-absence-of-take.

To make this adjustment to survival to account for the effect of take, only the *proportion* of deaths due to watercraft need be known. But, when forecasting the effect of a particular level of observed take, the *number* of deaths due to watercraft needs to be known. This requires consideration of carcasses that are never found. This proportion of carcasses not recovered has not been estimated, but is presumed to be in the range of 10-40% (B.B. Ackerman, *personal communication*). To reflect uncertainty in this parameter, a value for this proportion is randomly drawn from this range, for each replicate in the simulations.

The survival-rates-in-the-absence-of-take, so calculated, are applied in the manner described above (under “Environmental Stochasticity” and “Demographic Stochasticity”). Then the watercraft-related take is removed from the population by subtraction. The proposed take levels, because they are on the same scale as the observed take levels, are inflated to account for the proportion of carcasses not recovered and to account for the undetermined cause-of-death category.

In addition to this direct effect of watercraft-related take, an indirect effect of manatee-boat strikes will also be included in the model, to reflect a sublethal effect of boat strikes on reproduction (O'Shea et al. 2001). Breeding rates will be decreased as an increasing function of boat strikes, as indexed by the observed level of watercraft-related mortality. To develop this function, the breeding rates in the Northwest and Atlantic Regions will be compared to the per capita watercraft-related mortality rates in these two Regions. The differences in breeding rates will be assumed to be due to the differences in watercraft-related mortality rates. The implied relationship will be used in the model to reflect sublethal effects of boat strikes.

Simulation

There will be 10,000-100,000 replicates of 100-year simulations. Each simulation will begin with the 2001 minimum regional population sizes. Uncertainty about parameters will be handled by sampling once at the start of the 100-yr simulation and using those parameter values for every year of the simulation; environmental stochasticity will be handled by resampling the appropriate parameter each year of the simulation. The survival and reproductive rates will be calculated for each year, as a function of the proposed level of take (see below). Appropriate levels of sampling error will be included, based on the current precision of the existing surveys, in order to calculate confidence intervals. Each 100-year simulation will be summarized by the following metrics: for each region, were the recovery criteria met (yes/no) at 50 and 100 years; and for each region, when were the recovery criteria first met (i.e., what was the time to recovery)? From these values, aggregate measures can be calculated: were the recovery criteria met in all 4 regions at 50 and 100 years; and when were the recovery criteria first met simultaneously in all regions?

Three scenarios will be considered for each region: no take, negligible take, and no action. (1) Under the "no take" scenario, the observed watercraft-related mortality will be set to 0. This is the baseline scenario to which the others will be compared. (2) The "negligible take" scenario will assume that the level of watercraft-related mortality is held at some particular level. Many different levels will be examined to determine what meets the criteria for "negligible." This scenario will assume that the level of take will be held constant for the span of the simulations (100 yr), not just over the life of the regulation. (3) Under the "no action" scenario, watercraft-related take will be set at the levels recently observed, and will continue to increase at recent rates.

To evaluate the criteria for "negligible", the first two scenarios will be compared in parallel. The same values of the various random distributions will be used, so that each simulation without take is paired with a simulation with take. From each simulation without take, the same summary metrics will be calculated. By comparing the pairs (with and without take), a delay in the time to recovery can be calculated (as the ratio of the recovery time with take to the recovery time without take).

Finally, the 10,000-100,000 replicates (the number will depend on computation time available) will be summarized to calculate (a) the probability of recovery in 50 yr; (b) the probability of recovery in 100 yr; and (c) the probability that the delay in time to recovery is $\geq 10\%$. Note that

these metrics are associated with a particular set of proposed levels of take for each region. They will also be calculated for the “no take” and “no action” scenarios.

This entire process will be repeated with other proposed levels of take in each region. Adaptive search methods will be used to find the combination of allowable takes in each region that (a) maximize the total allowable take statewide, and (b) meet the criteria for negligible impact.

Various regional strategies will be compared as well. For instance, it will be possible to apply the “no action” scenario to the Northwest and Upper St. John’s regions, while applying the “negligible take” scenarios to the Atlantic and Southwest regions. Since the criteria for “negligible” are based on meeting recovery goals statewide, the finding of “negligible” applies to a statewide strategy, which may involve different scenarios in the different regions.

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